Realistic expectations of timing between conservation and restoration actions and ecological responses

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Private landowners, citizen groups, local communities, and governmental agencies invest enormous effort, time, land, and money into practices designed to conserve or restore ecosystem functions and structure. A recent survey estimated that river restoration in the United States amounts to more than $1 billion annually (Bernhardt et al., 2005a). In 1995 alone, federal expenditures on watershed-based programs to reduce agricultural pollution were estimated to exceed $500 million (General Accounting Office, 1995). Even though restoration costs are considered high by much of the public and local decision-makers, ecological benefits derived from those efforts are believed to exceed conservation and restoration expenditures (Costanza et al., 1997). For example, a study on a 72-kilometer (45-mile) reach of the Platte River estimated households along the river valued ecosystem services (water quality, soil erosion control, habitat, recreation) delivered at $19 million to $70 million annually, substantially more than the costs of conservation measures undertaken [e.g., water leasing at $1.1 million and Conservation Reserve Program (CRP) contracts of $12.3 million] (Loomis et al., 2000).

The needs, locations, and costs of conservation and restoration are constantly debated—always with passion, sometimes with information. An element frequently missing from these discussions is any realistic estimation of the time required before desired outcomes are attained (Stanford et al., 1996; National Research Council, 2002). While conservation or restoration actions are well-intended, expectations about timing of outcomes and effectiveness of such actions are often unrealistically short. As Wayne Elmore, a rangeland management scientist, noted, “Instant gratification is not fast enough for most Americans.” Our objectives here are to identify timeframes over which conservation and restoration outcomes in agriculturally dominated landscapes are likely to be realized; explore landscape, ecological, and social factors affecting the definition of success for these practices; and address how conservation policies can be designed, implemented, and evaluated to yield reasonable measures of the effectiveness of these practices in agricultural ecosystems.

Conservation versus restoration

Conservation and restoration are closely related but distinct processes. Dissimilarity between these concepts has enormous consequences in terms of how success of ecological responses to management actions is defined. Conservation attempts to maintain or protect functional and ecological components of ecosystems to sustain existing resources. In contrast, restoration attempts to repair ecosystem processes and components to restore functions or structure that have been impaired or eliminated. Restoration outcomes range from minor renovation of ecological processes to attempts for complete recovery of ecosystem structure and function, which is rarely attained. Ideally, conservation maintains the performance of the existing system. Depending upon the amount of degradation and degree of recovery possible, restoration may require decades or longer to realize measurable responses. In terms of realistic expectations, one of the most critical distinctions is that conservation attempts to protect existing ecosystem structure and function; desired outcomes can thus be achieved more immediately. But a major question that must be addressed is the degree to which responses from these practices can be maintained. In contrast, restoration practices are designed to restore a portion of impaired ecosystem structure and function; thus, desired outcomes may require decades or centuries before restoration goals are realized.

In addition to substantial time lags in ecological responses potentially associated with restoration, the spatial extent and location of restoration may lead to distinct ecosystem responses. As implied by the river continuum concept (Vannote et al., 1980), this is especially true for discharge-dependent characteristics, such as flow regime, water temperature, and water chemistry. The river continuum concept suggests the relative influence of riparian shading and allochthonous inputs should decline as rivers increase in size because (1) channels generally become wider with reduced area of effective shading, (2) the amount of allochthonous riparian carbon is dwarfed by autochthonous instream carbon, and (3) increases in the volume of water passing through any particular cross-section require greater inputs of energy or carbon to significantly alter water temperature or allochthonous carbon concentrations. The river continuum concept can be used when scaling expectations of ecosystem response to restoration. For example, measurable impacts of riparian restoration at a given location on water temperature or solute concentrations should only be expected if the restored system shades the channel for a substantial fraction of its sun-exposed length or intercepts a substantial portion of dissolved pollutants. Following this reasoning, the impact of a given restoration effort, as well as the ability to detect effects, depends upon the size of the area targeted for restoration. Larger, more spatially complex areas will require greater amounts of restoration effort to achieve similar levels of recovery than can be expected within smaller areas presenting less physically and ecologically intricate challenges.

Ecological restoration: Successes and failures

Water temperature

Surface water temperature is determined by many variables, but major factors influenced by human activity include water quantity, channel morphology, subsurface exchange, and riparian vegetation (Independent Multidisciplinary Science Team, 2004; Poole and Berman, 2001). Agricultural practices potentially alter all four factors leading to increased rates of thermal alteration (warming and cooling) along stream and river networks. Restoration actions related to water quantity generally focus on reducing withdrawals from surface waters, increasing efficiency of water use, and restoring groundwater sources. Recovery of channel dimensions along streams and rivers in agricultural land commonly requires restoration of riparian plant communities, management of livestock grazing, and reversal of stream channel incision processes. Restoration of subsurface exchange, either hyporheic or groundwater, includes reconnection of hydrologic flow paths (Younus et al., 2000; Ebersole et al., 2003) or restoration of depleted alluvial sediments. Recovery of riparian shade is one of the most common agricultural restoration efforts and includes replanting, natural regeneration, livestock management, and changes in land use (Marsh et al., 2005). The hydrologic, geomorphic, and ecological processes involved in restoration actions require differing amounts of time to achieve their goals. Most require decades at the very least. None can provide immediate recovery of stream temperature and its influence on aquatic ecosystems.

Evidence of temperature response to modification of riparian vegetation in agriculturally dominated basins suggests that removal of riparian vegetation increases stream warming while reestablishment of riparian shade leads to reduced warming (Wehrly et al., 1998; Independent Multidisciplinary Science Team, 2004; Wang et al., 2003). The influence of riparian shade on rates of warming diminishes as streams become wider and discharge increases. But a few studies have noted that shade has little or no influence on stream temperature where subsurface inputs are significant (Mosley, 1983), stream water temperature is similar to air temperature (Borman and Larson, 2003), or in large streams where the relative influence of shade on surface water area is minor (Bartholow, 1995). The overwhelming number of studies of wadeable streams, however, concludes that shade influences stream temperature, thus restoration of riparian vegetation may reduce rates of warming and observed stream temperatures (Independent Multidisciplinary Science Team, 2004; Wehrly et al., 2003). Therefore, a portion of stream temperature recovery requires reestablishment of canopy cover over the stream channel. Reestablishment of channel dimensions through riparian recovery may also lead to lower stream temperatures. Reestablishment of effective vegetative canopy cover generally requires 10 to 30 years, depending upon the size of the stream and the type of riparian plant communities restored.

In northern California, late-seral riparian forests maintained summer water temperatures supporting cold-water amphibians and salmonids, while streams in grasslands exhibited higher temperatures (Welsh et al., 2005). Another study in California concluded abundance and distribution of riparian canopy substantially influenced stream temperature in basins up to approximately 75,000 hectares (158,000 acres) (Lewis et al., 2000). Deforestation in Japan resulted in loss of riparian forests and increased maximum temperatures from 22 Celsius degrees (72 Fahrenheit degrees) to 28 Celsius degrees (82 Fahrenheit degrees) during a 50-year period (Nagasaka1 and Nakamura, 1999). Fish communities in Japan were strongly affected by temperature, with more salmonids in forested reaches than found within grassland reaches (Inouel and Nakano, 2001). Studies in New Zealand observed that removal of riparian vegetation by cattle increased stream temperatures 3.9 Celsius degrees (7.8 Fahrenheit degrees) to 7.8 Celsius degrees (14 Fahrenheit degrees) and altered the macroinvertebrate community structure (Quinn et al., 1992). Investigations of livestock grazing in eastern Oregon found streams with canopy covers greater than 75 percent supported water temperatures meeting thermal requirements for rainbow trout and Chinook salmon. The lowest temperatures were observed in streams without streamside grazing (Maloney et al., 1999). Grass-dominated riparian buffers can provide as much shade as buffers dominated by woody vegetation in small Minnesota streams, but wooded buffers exhibited the lowest maximum stream temperatures (Blann et al., 2002).

Water chemistry

The Chesapeake Bay watershed represents successful coordination among various local, state, and federal agencies, as well as an instructive lesson about expectations from efforts to manage nutrient discharges from urban and agricultural landscapes. Because agriculture is the single greatest source of nutrients in the Chesapeake Bay, significant efforts were directed toward reducing nonpoint-source nutrient inputs into the watershed. Early restoration efforts focused on erosion-based best management practices (BMPs); these were relatively successful at reducing particulate phosphorus losses from agricultural land, but less successful at reducing nitrogen, which is more often transported as dissolved nitrate (Boesch et al., 2001). Most efforts were process-based, however, focusing on landowners developing and implementing nutrient management plans. Reductions in nutrient loads resulting from those plans were typically assumed rather than directly assessed through monitoring of water quality. Although ambitious water quality monitoring programs were able to describe trends at the outlets of major tributaries, it was difficult to discern the causes when restoration activities failed to meet expected objectives. Further analysis suggested increases in annual rainfall during the past decade and time lags associated with dissolved transport in groundwater have occasionally contributed to elevated inputs in surface water, despite improved nonpoint-source nutrient management, further complicating an understanding of restoration efforts (Boesch et al., 2001). Synthesis of results from restoration projects in the Chesapeake Bay watershed (Hasset et al., 2005) suggest that, although the vast majority of restoration has focused on water quality or riparian management, relatively few projects have incorporated follow-up monitoring to assess water quality and ecological benefits. Therefore, it is difficult to evaluate the success of those restorations and their effectiveness in improving water quality in the Chesapeake Bay.

Pollutants and wastes

Characteristics of soils and sediments influence time lags between implementation of management actions and improvement in water quality. If the phosphorus content of soils is high, ceasing the application of manure or fertilizer will eliminate further increases, but many crops must be grown before soil test phosphorus declines to acceptable levels (Read et al., 1973; Halvorson and Black, 1985). As long as phosphorus in soils remains high, the soil will remain a source of particulate and dissolved phosphorous for transport to surface waters. Consequently, the impact of limiting phosphorus applications may not be immediately apparent. Phosphorus can also accumulate in wetland, streambed, and lake sediments. Sediments are a recognized source of phosphorus in the overlying water column and are implicated when the phosphorus content does not decline in proportion to a reduction in inputs (Marsden, 1989). The release of phosphorus from sediments does not occur at a constant rate because of the influence of sediment type, temperature, pH, redox potential, nitrate concentration, and physical disturbance (Holdren and Armstrong, 1980; Jensen and Andersen, 1992). In addition, time lag of ecological response to a conservation measure varies in response to specific environmental conditions. In an example from Washington, Lake Sammamish failed to show an ecological response to a one-third reduction in phosphorus loading for more than 10 years before improving markedly in the subsequent five years (Welsh et al., 1986).

Similarly, soils and sediments can amass pesticides that can contaminate water and impact the ecosystem long after applications have ceased (U.S. Geological Survey, 2006). Continuing detections and impacts of DDT and its metabolites years after discontinuance of use are examples of time lag for response to intervention. Long-term existence of pesticides in stream sediments is greatest for pesticides with little affinity for water (low solubility), but pesticides with relatively high solubility and relatively fast soil degradation rates have also been observed to persist in wetland substrates (Elliott et al., 2001).

Manures applied or excreted on agricultural land contain pathogens that can be transported to surface waters and deposited into sediment (Collins et al., 2005; Muirhead et al., 2006). Although pathogens are not as likely to persist as long as some pesticides, *E. coli* have been observed to survive and even exhibit temporary growth in freshwater sediments in laboratory experiments (LaLiberte and Grimes, 1982). *Escherichia coli* have also been shown to survive for up to six weeks in stream sediments and become resuspended in the water column during storm events (Jamieson et al., 2005).

Veterinary pharmaceuticals are present in manures applied to agricultural land, as witnessed by growing numbers of reports documenting detections of antibiotics and pharmaceuticals in streams and rivers (Koplin et al., 2002; Lindsay et al., 2001) and with clear indications that at least some originate from agricultural operations (Calamari et al., 2003). It is likely that pharmaceuticals will behave similarly to pesticides, possibly remaining is sediments long after their initial introduction to surface water. Diaz-Cruz et al. (2003) reported detections of veterinary drugs in sediments, and Halling-Sorensen et al. (1998) described the presence of persistent antibiotics in sediments of fish-farm sites where antibiotics had been administered.

Given the storage capacity of sediments for nutrients and contaminants, it is unrealistic to expect management alterations that reduce inputs will have an immediate impact on water quality. Even drastic actions, such as the elimination of all pesticide applications, may not reduce concentrations to levels that can be explained by atmospheric transport until the legacy of past pesticide applications remaining in sediments are depleted. Consequently, it is important not to celebrate an apparent success prematurely because pesticides may temporarily disappear from the water column, only to reappear as they are released from sediments (Cessna and Elliott, 2004).

Some management practices have inherent time lags between establishment and their expected environmental response. For example, conservation tillage has been found to reduce soil erosion 70 percent or more in upland areas, but monitoring programs often fail to detect significant reductions in sediment loss at the watershed outlet for a decade or more. This may be the result of a temporary increase in gulley and channel erosion or large quantities of sediment already in storage at the watershed level. Until the channel system reaches a new hydraulic equilibrium with reduced sediment inflows, the sediment that once came from upland areas will be replaced by sediment from channel erosion.

Similarly, establishment of riparian buffers may disturb streambanks and have a temporary negative impact on water quality. Several years may be required for vegetation to become sufficiently established for the buffer to become effective. Conversely, the effectiveness of nutrient removal by an established buffer often declines over time as nutrients accumulate in flow paths (Sheppard et al., 2006).

Climatic effects on hydrology and water quality often have greater effects than could be expected, outweighing environmental responses from a conservation practice (Maulé et al., 2005; Glozier et al., 2006). Simultaneous monitoring of weather and water quality may allow detection of subtle changes due to management that may otherwise be undetectable. Another approach is to examine event hydrographs and only compare pre- and post-management water quality for hydrologically similar events (Glozier et al., 2006). Nonetheless, it should be expected that many years of monitoring data will usually be required to separate conclusively management effects from those affected by climatic variability.

While some ecosystem impairments take decades to recover, other watersheds may respond quickly to conservation measures. Water quality impairments caused by nonconservative contaminants, such as bacteria from human and livestock sources, which die-off or degrade quickly in the environment, have been quickly reduced in some cases. A case in point is the North Fork River total maximum daily loads (TMDL) (U.S. Environmental Protection Agency, 2004). In 1998, a bacteria (fecal coliform) impairment TMDL was developed for the 806-square-kilometer (311-square-mile) watershed in West Virginia. Both point and nonpoint bacteria sources were identified with pastureland, failing septic systems, and direct in-stream deposition via cattle defecation identified as the primary causes of bacterial impairment. The TMDL required a 36 percent load reduction from agriculture and pastureland and no reduction from other sources.

In 1998 the U.S. Department of Agriculture's Natural Resources Conservation Service (NRCS), the Potomac Valley Conservation District, and the North Fork Watershed Association began work on a management plan to lessen damage from flooding and improve water quality within the watershed. In 2000, the North Fork Watershed Association obtained U.S. Environmental Protection Agency (EPA) 319 funding to implement the management plan. Implemented BMPs included fencing along streambanks, alternative livestock watering facilities, livestock water wells, riparian buffers along streams, nutrient management plans, educational programs, manure and poultry litter composting, and stream restoration. Approximately 85 percent of the farmers in the watershed were actively involved in implementing voluntary, incentive-based BMPs. The North Fork River was delisted for the fecal coliform impairment in 2004, based upon monitoring data collected from 1998-2000 showing that BMPs can effectively address water quality issues.

The effectiveness of riparian buffers, filter strips, and similar practices in reducing pollutant loadings from agricultural land has been heavily researched, but remains poorly understood, with results believed to be extremely site specific. In a review of 72 journal articles, 59 published since 2000, dealing with primary research on the effectiveness of buffers for water quality protection, buffer efficiencies were reported to be relatively high (Table 1).

Unfortunately, reported efficiencies, such as those shown in table 1, may not be representative of real-world buffer efficiencies because most experiments poorly represent field conditions and/or the long-term effectiveness of buffers. Most experimental studies have four serious limitations that constrain effectiveness in representing field conditions:

1. Most buffer research is conducted on small plots constructed so that shallow, uniform flow across the plots is maximized. In the real world, shallow, uniform flow is the exception, and most flow from upland areas crosses buffers as concentrated flow, which greatly reduces buffer effectiveness. Thus, experimental studies that do not consider concentrated flow effects tend to overestimate buffer effectiveness.
2. Most buffer research is conducted on small plots with small source-area-to-buffer-area ratios that are not representative of buffers installed under actual field conditions. For example, across the 69 studies analyzed in table 1, the source-to-buffer area ratio ranged from 0.4:1 to 55:1, with a median of 5.5:1. This median value would require the conversion of 18 percent of agricultural land to buffers and is considerably higher (two to three

Table 1. Reported effectiveness of riparian buffers for reducing nonpoint-source pollutants (runoff, sediment, nutrients, and pesticides).

	Reduction		
Parameter	**Range (%)**	**Mean (%)**	**n**
Runoff	21 to 88	51	8
Biological oxygen demand	18	18	1
Ammonium	28 to 87	65	9
Nitrate (runoff)	9 to 99	69	13
Nitrate (subsurface)	49 to 91	72	6
Phosphate	36 to 98	73	8
Total Kjeldahl nitrogen	11 to 79	48	5
Total nitrogen	37 to 94	64	11
Total phosphorus	5 to 91	61	18
Sediment	17 to 100	84	69
Atrazine	22 to 70	51	6
Metolachlor	51 to 66	56	3
Fecal coliform	28 to 90	67	4

times) than the recommended, or allowed, ratio in most buffer programs.

3. Most experimental buffer studies are conducted on newly established buffers (typically less than a year since establishment), with most monitoring lasting for less than a month. Thus, most experimental buffer study results represent effectiveness only during establishment, failing to furnish estimates of long-term effectiveness.
4. Most experimental buffer studies on cropland either use or simulate conventional tillage and agrochemical applications in the experimental area. This is unrealistic. Buffers should only be used in concert with in-field systems of conservation practices designed to keep sediment and agricultural chemicals in the field where they are valuable resources rather than pollutants that need to be trapped by buffers (Dillaha et al., 1989). The few buffer experiments simulating high sediment and nutrient loadings over longer time periods suggest the effectiveness of overloaded buffers will decline dramatically over time.

Only one of the studies summarized in table 1 (Udawatta et al., 2002) simulated real world conditions in terms of concentrated flow patterns, reasonable source-to-buffer-area ratio, and use of infield conservation practices (no-till) in addition to buffers. This three-year study used a paired approach, with a control watershed in row crops [1.6 hectares (4 acres)] and two treatment watersheds: One with grass buffer strips [3.2 hectares (7.9 acres)] and the other with trees in grass buffer strips [4.5 hectares (11 acres)]. No-till was used on cropland in all three watersheds. Grass buffers and trees in the agroforestry treatment were established in 1997, with monitoring initiated at the same time. The buffers consisted of a system of in-field contour buffers and grass waterways along major in-field drainageways. The cropland-to-buffer-area ratio was approximately 8:1, with about 13 percent of the treatment watershed area devoted to buffers. Runoff, sediment, and nutrient losses were monitored at watershed outlets. The control watershed had total phosphorus, total nitrogen, and nitrate losses of 0.42, 1.52, and 0.28 kilograms per hectare per year (0.92, 3.36, and 0.63 pounds per acre per year), respectively, indicating no-till was effective in minimizing nutrient losses without buffers. The grass buffer and agroforestry treatments reduced surface runoff

10 percent and 1 percent, respectively; sediment losses increased 35 percent and 17 percent, respectively; total phosphorus losses declined 8 percent and 17 percent, respectively; total nitrogen losses declined 14 percent and 11 percent, respectively; and nitrate losses declined 21 percent and 5 percent, respectively. The reported increases in sediment losses with the buffers were unexpected, but the losses with the source areas in no-till were so low that the increase was negligible. Sediment losses from the control, grass buffer, and agroforestry treatments were 27, 33, and 36 kilograms per hectare per year (60, 72, and 79 pounds per acre per year) over the three-year study, which is extremely low and indicative of excellent no-till production.

Aquatic communities

The extent of actions intended to improve aquatic habitats in agricultural landscapes varies across the United States and Canada because agricultural land is generally privately owned and management objectives may not include concern for fish and wildlife habitats. While U.S. Department of Agriculture (USDA) farm bill programs offer increasingly attractive financial incentives for conservation of aquatic resources, the degree to which restorative actions are implemented and monitored for effectiveness is challenging to evaluate and report. This is apparent by the poor rate at which restoration projects have been evaluated (Bernhardt et al., 2005a). This lack of evaluation is a consequence of limited dollars allocated for monitoring and a failure by those who formulate conservation policies to recognize the importance of long-term monitoring to refine performance of conservation programs and practices. Monitoring designs are necessarily intricate and expensive to implement because of the ecologically complex nature of stream, river, floodplain, and upland processes. Nevertheless, monitoring is essential to determine what works and does not work under different circumstances and to gain knowledge on how long it takes for conservation practices to become effective. In a blue ribbon panel's review of USDA's Conservation Effects Assessment Project (CEAP) (Soil and Water Conservation Society, 2006), panel members concluded that lack of resources for monitoring was a significant limitation of CEAP and other conservation programs. They concluded: "The most important and troubling missing piece is the absence of plans for on-the-ground monitoring of change in the environmental indicators and outcomes conservation programs and activities are intended to improve." The panel recommended that Congress mandate that at least one percent of the funding for each authorized program—about $40 million of the $4 billion U.S. taxpayers are investing in conservation— be set aside to support monitoring and evaluation of those programs.

Restoration actions targeted to improve habitats for aquatic species are difficult to evaluate because effects can be influenced by physical, biological, and chemical responses at multiple spatial and temporal scales having variable affects on biological communities (Minns et al., 1996; Lammert and Allan, 1999; Fitzpatrick et al., 2001; Vondracek et al., 2005). Moreover, suites of practices installed either sporadically or strategically in a catchment will differentially influence the breadth and timing of response of stream or wetland species and their physical habitats. Thus, correlations between a specific practice and the ecological response of an organism or its habitat are not easily discerned. These limitations aside, recent studies focusing on effects of agricultural practices on conservation of aquatic species and their habitats are beginning to offer insights into which practices may be effective at arresting declines in North American aquatic species. In most cases, management practices that retain or improve connections among ecological processes and/or different aquatic habitats contribute to the quality of those habitats and the well-being of the aquatic species that inhabit them.

Along stream and river corridors, fish, amphibians, and aquatic insects move among different habitat types, including pools, riffles, backwaters, wetlands, sloughs, alcoves, hyporheic zones, and riparian zones during their life cycles. Agricultural practices can be modified to maintain connections between essential components of habitat across space and time. In 20 streams in agricultural land within the Minnesota River Basin, wooded riparian areas supported higher fish richness, diversity, indices of biotic integrity, and macroinvertebrate communities than recorded within nonwooded, open reaches (Stauffer et al., 2000). Restoration practices that effectively reconnect upstream and downstream aquatic habitats include providing fish passage around or through barriers, such as dams or poorly constructed culverts (Pess et al., 1998; Hart et al., 2002; Johnson, 2002). Breaching dikes potentially reconnects riverine migration routes with estuarine rearing and holding habitats (Frenkel and Morlan, 1991). Installation and active management of water control structures in constructed or restored wetlands have been effective in preventing entrapment, allowing fish to emigrate out of floodplain wetlands entered during seasonal high flows (Swales and Levings, 1989, Thomson et al., 2005; Henning, 2005).

Keeping fish and water in streams and out of irrigation ditches increasingly is an objective of ranchers and farmers in the arid west, triggering installation of sophisticated fish screens for irrigation diversions (McMichael et al., 2004) and effective irrigation conservation management techniques through candidate conservation agreements [David Smith, USDA-NRCS, personal communication: (http://www.mt.nrcs.usda.gov/about/mtstcm/feb05/grayling.html).]

Simply maintaining physical connectivity between intermittent stream channels used as drainage ditches and mainstem rivers has been shown to influence the amount of winter habitat for native fish, benthic invertebrates, and amphibian species in the grass seed farms of the Willamette Valley of Oregon (Colvin, 2005). Similarly, maintaining open drains on agricultural land in Ontario provides habitat for fish assemblages identical to those inhabiting nearby streams (Stammler, 2005).

Connecting habitats includes maintaining ecological linkages between riparian zones and streams. For example, riparian vegetation structure influences the composition and abundance of terrestrial insect communities. By altering grazing management regimes to favor persistence of riparian vegetation where terrestrial insects thrive, fish benefit from seasonally important food sources. Grazing systems that allow cattle to graze for short durations increase terrestrial insect production, which has been shown to correlate strongly with fish condition and survival on Wyoming ranchland (Saunders, 2006; Saunders and Fausch, 2006).

Loss of cropland due to streambank erosion has elevated interest in riparian management that includes replanting of herbaceous and woody riparian buffers, often coupled with instream rock or wood to deflect the flow away from unprotected banks. Preliminary investigations in western Oregon indicate such streambank stabilization practices, if designed correctly, encourage instream processes important to aquatic species, including retention of detritus and large wood for fish cover and macroinvertebrate food sources (Stan Gregory, unpublished data). Studies in Minnesota further support the importance of riparian corridor conservation and restoration to aquatic species because it contributes to instream habitat and geomorphic features at multiple scales (Stauffer et al., 2000; Blann et al., 2002; Talmage et al., 2002).

Instream structural improvements have improved fish habitats at some sites. Assessment of the effectiveness of instream structures placed in western Washington and Oregon streams over the last three decades revealed that the majority of sites exhibited significantly higher densities of juvenile coho salmon, steelhead, and cutthroat trout after restoration (Roni and Quinn, 2001). While placement of instream log structures has proven valuable in the Northwest, failures in the effectiveness of this practice in the southeastern United States indicate re-introduction of large wood to drastically altered stream systems is often unsuccessful when placed in stream reaches physically unable to retain them (Shields et al., 2006).

Terrestrial wildlife

The purpose of USDA conservation programs is not to restore native ecosystems but to lessen undesirable environmental effects of agricultural production. Yet these policies, at times imperfect, have brought about significant improvement in the quality and distribution of wildlife habitats associated with agricultural land across much of the American landscape. Fundamental to the design of successful conservation and agricultural policies is recognition that farming and environmental quality improvements are not mutually exclusive goals, nor are environmental solutions associated with soil erosion, water quality, and wildlife habitats independent issues.

Established in 1986, embedded within all 50 states, and composed of an eclectic mix of conservation practices, the 14.6-million-hectare (36-million-acre-plus) CRP represents a cornerstone of USDA conservation policy. Investigations describing the environmental, social, and economic effects of CRP offer insight on at least some effects of conservation policies on wildlife and their habitats (Allen and Vandever, 2005; Haufler, 2005). Some benefits have been profound, such as 25 million ducks produced in the Prairie Pothole region due to the nesting cover provided by CRP grassland. Other benefits are more understated—doubling of the range of mule deer across the Texas Panhandle, for example, or the reversal in population declines of various songbird species in response to CRP grassland replacing crops on highly erodible land. Many CRP conservation practices (e.g., planting of native and introduced grasses, field borders, riparian buffers) are implemented in other federal and state conservation programs. It seems reasonable to assume wildlife-related effects described for individual CRP conservation practices have similar benefits and consequences when applied as part of these other programs as well.

Economic and social support for rural communities, aesthetically pleasing landscapes, recreational opportunities, and sustainable populations of wildlife represent ecosystem services delivered from agricultural land use whose importance is not often adequately captured in assessments (Feather et al., 1999; Costanza et al., 2000). Although wide-ranging personal and social effects of the CRP remain impractical to measure, these nonquantifiable benefits are valued particularly by those most directly affected. CRP participants attribute improving future productivity of land, retention of water from rain and snow, reappearance of springs, improved quality of well water, prevention of unwanted urban expansion, stability in income, lower operational costs, and control of drifting snow as program benefits (Johnson and Maxwell, 2001; Bangsund et al., 2002; Allen and Vandever, 2003). For many, the CRP has enhanced aesthetic qualities of their farmland, brought greater numbers of wildlife, and increased opportunities for recreational and social use of their land. Many of these benefits were delivered soon after establishment of conservation practices, but an accurate assessment of their economic and social significance remains elusive.

For the sake of simplicity, visualize most wildlife inhabiting agriculturally dominated regions as belonging in one of two groups. Farmland wildlife (e.g., ring-necked pheasant, bobwhite quail, white-tailed deer) generally prosper where a relatively small proportion (e.g., less than 10 percent) of the landscape is dedicated to nonfarmed vegetation, with crop production remaining the prevailing land use. The other category can be characterized as wildlife endemic to grassland (e.g., upland nesting waterfowl, prairie chickens, and pronghorn antelope). These species are generally dependent upon relatively large, contiguous blocks of grassland cover. Farmland species benefit from high levels of interspersion between farmed and nonfarmed land uses; most wildlife species endemic to grassland ecosystems do not.

Conservation programs administered by USDA have benefited species whose elemental habitat requirements are met by conservation practices designed to most appropriately address regionally prevalent forces of soil erosion. In drier, western regions, whole fields planted to grasses offer the greatest opportunities to address wind erosion and the needs of grassland wildlife. In wetter climates, where soil erosion by water is an issue of greater concern, grass filter strips, riparian buffers, field borders, and removal of smaller tracts of erodible land from cultivation typically enhance habitat quality for farmland wildlife adapted to higher levels of interspersion between land uses.

Time lags in ecological responses

Restoration practices inherently require variable periods of time for ecological processes to deliver desired outcomes, for systems to adjust to restoration measures, for invasive species to be reduced, for desirable endemic species to increase, for toxicants and other forms of degradation to be eliminated or isolated in long-term storage, and for connections between habitats, communities, and ecosystems to be restored (Harding et al., 1998; Sarr, 2002; Bond and Lake, 2003). Wetland restoration studies in the southeastern United States found recovery of amphibian communities was affected by drought and disease after seven breeding seasons, leading to the conclusion that long timeframes are necessary for monitoring programs to assess accurately the outcomes of restoration practices (Petranka et al., 2003). Time lags in replanted vegetation reaching maturity were identified as one of the most serious limitations of restoration for birds and arboreal mammals in Australian agricultural landscapes (Vesk and MacNally, 2006). Monitoring vegetative characteristics of CRP grassland in the Great Plains over a 12-year period, Cade et al. (2004) found that vegetative variables affecting the quality of wildlife habitats varied not only by grass species planted, but also through time and in response to natural or human-induced disturbance. Harding et al. (1998) concluded that the best predictors of present macroinvertebrate communities in streams of the southeastern United States were land use and land cover conditions in the 1950s. The influences of past agricultural land uses on invertebrate communities were still evident after more than 45 years, even though the local riparian areas had become reforested. As Bond and Lake (2003) noted, "…legacies of past disturbances and the impacts of on-going disturbances operating at larger (possibly catchment-wide) scales can compromise works done at individual sites or reaches."

Temperature

Restoration of thermal regimes in stream networks is dependent upon processes that influence shade, discharge, channel dimension, and hyporheic exchange. Restoration of riparian shade clearly requires many years for an adequate, contiguous vegetative canopy to develop along a reach. Geomorphic processes may require decades to adjust channel dimensions, and reconnection of hydrologic flow paths for subsurface exchange are functions of channel structure and hydrologic regimes. A New Zealand study compared physical and biological characteristics of nine riparian buffers, replanted and fenced between 2 and 24 years, with conditions found within control reaches (Parkyn et al., 2003). Some stream properties, such as water clarity and channel stability within treated reaches, responded rapidly. Other characteristics, such as nutrient concentrations and presence of fecal coliform bacteria, were highly variable. Macroinvertebrate community composition did not respond within the time period investigated, which was attributed to the lack of response in stream temperature. Stream temperature could not be expected to adjust until canopy cover by riparian vegetation had recovered (Quinn et al., 1992).

Past or future changes in hydrologic connections can affect the location and timing of thermal responses to restoration. Roads, ditches, and diversions can also influence stream temperatures by changing the routing of surface and subsurface flows, which may be warmer or cooler than the stream temperature (Story et al., 2003). Consequently, stream temperatures may not respond to recovery of riparian vegetation if the routing of water from ditches or drains significantly alters stream temperatures. Also, restoration of hydrologic connectivity and detention through recovery of hyporheic zones through channel aggradation or reestablishment of wetlands may require several years or decades for hydrologic paths to become reestablished and well integrated into the flow network.

Nutrients and contaminants

Groundwater nitrate, leached from surface soils via subsurface flow to near-stream zones, may be an important source of nitrogen to surface waters (Cirmo and McDonnell, 1997). In some river systems, groundwater can make up as much as 50 percent of river flow, and groundwater may be decades to centuries older than surface water (Michel, 1992). In such systems the potential for time lags in water delivery can have a profound impact on the ability to detect degraded systems and quantitatively describe responses to restoration. For example, if recent land use practices lead to eutrophication of surface waters, it is possible for dilution by older and deeper flow systems with higher quality water to mitigate observed water quality degradation, particularly during baseflow conditions. On the other hand, shallow groundwater can retain nitrate concentrations for 40 years or more in the absence of reducing sediments (Bohlke and Denver, 1995). In these systems, detecting positive effects in post-restoration monitoring can be hampered by delivery of enriched, pre-restoration water to the stream. In such flow systems, however, long time lags reflect slow rates of delivery; hence, the ability of deeper flow systems to influence instantaneous stream concentrations would require substantial groundwater sourcing. Thus, although the full benefits of restoration practices may be masked in some systems by lags imposed on nitrogen-enriched groundwater, significant masking after a decade should be unusual.

In Mid-Atlantic States, nitrogen leaching from tributary watersheds of the Chesapeake Bay has increased since 1985 despite widespread restoration activity (Lindsey et al., 2003). Although patterns of individual watershed discharges vary, there is no clear trend across the basin (Alexander and Smith, 2006), leading to concerns about the effectiveness of nearly 20 years of restoration efforts under Chesapeake Bay agreements (Boesch et al., 2001). One recent study showed that although base flow is made up of water between 1 and 50 years old most water in the Chesapeake Bay watershed enters streams within a decade (Lindsey et al., 2003). Although the proportion of baseflow in streams can be influenced by the quantity of annual precipitation, average residence times for groundwater range from 10 to 20 years (Michel, 1992; Focazio et al., 1997). A comparable range of 2 to 9 years has been observed for nitrogen concentrations in waters at the Mississippi River outlet (McIssac et al., 2001), as well as 5 to 10 years for large rivers in Latvia (Stalnacke et al., 2003).

The time lag in nitrogen recovery introduced through soil percolation and groundwater contribution to surface water is relatively short compared to the lag expected in phosphorus recovery due to percolation pathways (Oenema and Roest, 1998). In soils with low phosphorus sorption capacities, unsustainable additions lead to soil saturation, and thereafter phosphorus concentrations in groundwater will increase with the degree of phosphorus saturation. Under these conditions, conservation actions that act to reduce or eliminate phosphorus application surpluses will have no immediate impact on phosphorus reaching surface waters by the percolation pathway. Model estimates suggest phosphorus transport through surface pathways may respond within 5 to 50 years, but phosphorus moving by the percolation pathway may take centuries to respond to management changes (Schippers et al., 2006).

Besides limiting observed benefits of restoration, knowledge of subsurface flow pathways can increase understanding about effectiveness of restoration activities. Molenat and Gascuel-Odoux (2002) showed that reduced nitrogen leaching along a 500-meter (547-yard) field-to-stream transect with three distinct flow pathways lowered recharge nitrogen concentrations from 100 to 80 milligrams per liter (100 to 80 parts per million) while simulated stream concentrations declined from 57.4 to 45.9 milligrams per liter. Water lag times in this study ranged from less than one year to three years. By redistributing patterns of nitrogen leaching to take advantage of longer travel times and denitrification from pyrite-rich subsurface sediment layers, the authors achieved similar reductions in simulated stream concentrations without changing average groundwater loadings. Thus, in addition to clarifying understanding about the timing of restoration effects, knowledge of groundwater flow pathways can be used as a mitigation or restoration tool to help reduce stream nutrient concentrations (Lindsey at al., 2003).

The Walnut Creek monitoring project in central Iowa investigated response of stream nitrate concentrations to changing land use patterns in a 5,218-hectare (12,894-acre) agricultural watershed over 10 years (Schilling and Spooner, 2006). In 1990, soybeans and corn constituted 69 percent of land use in the Walnut Creek watershed. Between 1990 and 2005, land devoted to row crops declined from 69 percent to 54 percent of the watershed area as a consequence of a U.S. Fish and Wildlife Service prairie restoration project. As a result of the land use changes and implementation of nutrient management programs between 1995 and 2005, nitrogen applications in the watershed declined 21 percent. Nitrate concentrations, however, still exceeded the standard of 10 milligrams of nitrate-nitrogen per liter for drinking water, with concentrations highest in the spring and early summer. Over the 10-year monitoring period, trend analysis indicated nitrate concentrations declined by about 0.12 milligrams per liter per year, or a total of 1.2 milligrams per liter for the whole basin, and by 8 to 12 milligrams per liter in smaller subbasins if a control watershed was used as a covariate. Without adjusting for the control, the reduction was 0.07 milligram per liter per year for the overall basin. Schilling and Spooner (2006) had estimated that a 10 percent change in row-crop area was required for a 1.95-milligrams-per-liter change in nitrate levels over a 10-year period. The lag time between reduced applications of nitrogen fertilizer and nitrate levels in Walnut Creek was influenced by the mean residence time for groundwater, which was estimated to be 14 years. Consequently, Schilling and Spooner (2006) concluded that it was impractical to detect changes in nitrate water quality in larger watersheds in less than several decades, and documentation of improvements in water quality due to conservation practices should focus on small subbasins where changes can be detected in shorter time frames.

Another mechanism influencing efficiency of denitrification in riparian areas is hydrologic connection between enriched groundwater and biogeochemically active sediments (Hill, 1996). Results from investigations in a series of European riparian areas suggested differences as small as 20 to 30 centimeters (8 to 12 inches) in water table depth had a significant effect on denitrification rates (Hefting et al., 2004). Channel incision and/or ditching to improve field drainage are common in agricultural land use, though sometimes incision is an unintended consequence of increasing channel flows. Such hydrologic modification can result in disconnection between enriched groundwater and denitrifying soil layers. Thus, restoration success can be hampered both by changes in hydrologic routing that reduce exposure of nitrogen-enriched waters to denitrifying sediments and alteration of the redox conditions required for denitrification (Pinay et al., 2002). Across whole watersheds, lack of hydrologic connection between nutrient sources and streams can lead to poorly buffered systems, even when a substantial portion of near-stream zones are forested (Weller et al., 1998; Baker et al., 2006).

Aquatic communities

The challenges of detecting and describing ecological successes or failures in improving conditions for aquatic species are due to multiple factors, not the least of which is the inherent variability in life-history patterns of aquatic species. Because fish assemblages are variable from day to day, month to month, year to year, and longer periods, data collected at randomly selected sites to determine if fish are responding to habitat improvements are difficult to interpret (Adams et al., 2004). This challenge may, however, be less daunting than the conflict between time lags in responses of species, habitats, and landscapes and the essentially nonecological timeframes of human systems. Farm policy, political administrations, landowner dynamics, and agency personnel change many times before watersheds can demonstrate recovery. Legislators want proof that restoration actions are worth the money invested, yet scientists provide only scant amounts of data that often cannot unequivocally prove success in the timeframe demanded by those who formulate or fund legislation affecting conservation policies. Failure to recognize complexities of natural and managed systems, recognition of time lags after implementation of conservation practices, and the historical lack of funding in support of long-term monitoring programs are underlying causes limiting the ability of science to answer fundamental questions about effectiveness of conservation practices and policies on aquatic species. Dynamic systems, such as rivers and streams, change constantly in response to natural disturbances and human perturbations. Conservation policymakers need to recognize change is not only normal in ecological systems, but confounding. Existing environmental issues and unanticipated effects of land use have occurred over decades and centuries. In most cases, it is unreasonable to expect that conservation or restoration will have immediate and permanent benefits to aquatic species and their habitats.

On the other hand, it is quite reasonable to assume that changes in land use practices in uplands will influence the habitats of aquatic species because aquatic systems are a reflection of environmental conditions in a watershed. Conservation tillage, residue management, and conservation buffers that improve overall surface water quality will, over time, benefit the species that use surface waters as habitats. Similarly, where clear, cold water exists, coldwater species can likely exist. Thus, conditions that influence temporal changes in stream temperature (as described previously) also influence temporal species responses. Conservation practices, such as riparian buffers designed to provide shade and channel features that maintain coolwater refuges, will over time provide habitat for species in search of such habitats, assuming a population source exists and barriers do not restrain immigration to those habitats. Some restoration measures do result in an immediate response by fish. Studies in the Pacific Northwest demonstrate success in reconnecting migratory routes and their habitats for anadromous salmonids (Beamer et al. 1998) and providing cover (Roni and Quinn, 2001). Kanehl et al. (1997) evaluated removal of a low-head dam and determined that both stream habitat and desired fish assemblage improved within five years.

Terrestrial wildlife

Effects of conservation polices on wildlife may be seen in a relatively short period of time or may take years to yield observable results. Removal of environmentally sensitive land from crop production has brought observable and immediate benefits to some species, but effects of alternative production and conservation practices, such as minimum tillage and terraces, are not always obvious or quantifiable. The cumulative effects of these practices, however, contribute to improvements in the quality of aquatic habitats downstream from the fields where the practices are applied.

A majority of investigations describing CRP effects on wildlife and their habitats have been completed on the scale of individual fields or by conservation practice (e.g., riparian buffers). The presence of conservation features in isolation, however, rarely has a definitive influence on abundance and distribution of many wildlife species. Rather, overall land use, cropping practices, and the spatial configuration of conservation practices with land remaining in production define long-term capabilities of agriculturally dominated landscapes to support viable populations of wildlife (Rodgers, 1999; Krapu et al., 2004; Taylor et al., 2006). Specifically linking quantitative responses of wildlife with conservation practices depends upon the species in question and becomes complex because wildlife species respond differently as vegetative characteristics change through time and in response to application, or absence, of disturbance brought on by tillage, fire, grazing, or other management practices (McCoy et al., 2001; Fritcher et al., 2004; Cade et al., 2005). Individual conservation practices may be beneficial for one species, but have negative effects on the suitability of habitat for others. For example, in the Texas panhandle, mule deer have expanded their range into heavily farmed landscapes as a consequence of the cover provided by introduced species of grass under the CRP. The same conservation practice, however, has concurrently diminished availability of habitat for swift fox because the vegetation becomes too tall and unsuitable for the animal's use (Kamler et al., 2001; Kamler et al., 2003).

During the past two decades, there have been many outstanding studies on how wildlife responds to the inclusion of conservation practices in intensively farmed landscapes. These investigations have been, and continue to be, used to refine USDA conservation policies and management guidelines. Hard numbers or measures are needed through which progress toward specific goals can be measured. Wildlife management in agricultural landscapes is well described; however, it is difficult to predict how numbers or distributions of wildlife will change in response to conservation practices. The one overarching criticism that might be directed toward research into wildlife response to conservation policies within agricultural ecosystems is a lack of focus on specific species, making identification of precise, quantifiable goals difficult. If specific goals cannot be identified for unique areas (e.g., farm, watershed, region) it is impossible to furnish measures that accurately describe progress toward reaching those goals.

Wildlife response to contemporary conservation policies in agricultural landscapes is potentially diverse, but it is not possible to optimize management for all species. There are wildlife species whose abundance and distribution reflect a practical balance between conservation and economically viable agriculture. Across much of the Great Plains and Corn Belt, for example, the ring-necked pheasant is perceived as a symbol of balance between agricultural production, conservation, and social value. The same circumstance is represented by upland-nesting waterfowl in the northern Great Plains, across the Southeast by the bobwhite quail, and by anadromous fisheries and sage grouse in the Northwest. Grassland birds in the Northeast are also species that can stand as emblems of balance between agriculture and conservation. These are generally the species about whose habitat needs the most is known. If habitat for these species is furnished, the needs for many, not all, other wildlife species inhabiting agriculturally dominated landscapes will be provided. It is the known habitat needs of these species defined at the field, farm, and watershed levels that offer greatest potential to define beneficial management practices and measurable goals through which the effectiveness of conservation can be more precisely described.

Acceptance of conservation goals affecting wildlife habitat and environmental quality in agricultural landscapes presents social as well as scientific challenges. Conservation programs have been an important source of income for small, intermediate, and rural-residence landowners who are less likely to adopt practices requiring substantial economic investment, technical skills, or management-intensive alternatives (Lambert et al., 2006). Larger operators, whose primary occupation is farming, are more likely to dedicate a smaller percentage of their land to conservation, but they are more likely to install practices generally requiring higher costs and compatibility with sustainable production of crops. The desires and limitations of landowners with differing personal and economic goals must be a part of any successful effort to enhance wildlife habitats associated with agricultural land use over the long-term.

Measuring cumulative effects

In many ways, "cumulative effects" is a vague concept applied to complex interactions. Rigorous scientific assessment of cumulative effects most commonly addresses coupled processes that lead to complex outcomes, but often does not fully address the full range of collective effects. In many ways, the spatial, temporal, and social complexity of landscape-level cumulative effects far exceeds the capacity of most environmental measurement and analysis systems. Yet management of simple sets of processes or small numbers of target species often leads to overly simplistic conclusions and adoption of practices that may degrade other resources. Analyses of multiple factors and processes along river networks has provided important frameworks for restoration of stream ecosystems and associated riparian areas (Li et al., 1994; Gore and Shields 1995) that may be applicable for evaluation of other conservation and restoration practices within agriculturally dominated landscapes.

Cumulative effects of riparian buffers and nutrient responses

Although much effort has been focused on the benefit of riparian buffers and restoration at local sites, comparatively little work has addressed cumulative downstream impacts on water quality (Dosskey, 2001). Recent advances in use of stable isotopes seem promising (e.g., Bohlke et al., 2004), but few tools exist to distinguish permanent from temporary nitrogen sinks across whole watersheds and signal a definitive response to restoration. Because most agricultural land use patterns reflect aggregate land use decisions by individual landowners and most watercourses within watersheds are not well-buffered, it is difficult to detect and measure effects of restoration activities. Baker et al. (in press) recently studied land-cover patterns in more than 500 watersheds from four physiographic provinces within the Chesapeake Bay watershed. The authors compared watershed cropland proportions with proportions adjusted downward to represent presumed effects of existing riparian forests and wetlands. In this manner, they sought to examine whether extant patterns of riparian buffers were likely to result in reduced nutrient discharges compared to those expected from unbuffered areas. Results of the investigation led the authors to conclude that even when riparian buffers were assumed to reduce nutrient concentrations as effectively as in published studies (e.g., Lowrance et al., 1997) most watersheds showed buffer patterns that would not lead to a substantial reduction in nutrient discharges. This finding underscores the need for widespread changes in land use practices that include establishment of riparian buffers as well as the importance of multiple strategies for reducing nutrient exports.

Most studies of riparian buffers demonstrate water quality benefits measured along field-to-stream transects (e.g., Peterjohn and Correll 1984; Lowrance et al., 1997) or describe substantial denitrification potential (e.g., Groffman et al., 2002; Addy et al., 2002). By implementing buffer restoration, many managers assume the costs of restoration will be offset by the benefits described in the scientific literature. Prevailing evidence in the form of spatial and temporal variation in buffer effectiveness suggests, however, that the water quality benefits of any buffer restoration are likely to be conditional rather than universal (e.g., Jordan et al., 1993; Hill, 1996; Correll et al., 1997; Vidon and Hill, 2004; Hefting et al., 2004). There may be a wide range of water quality benefits achieved by placing restoration activities at specific locations (e.g., Dosskey et al., 2005), but at present, there is little coordination of restoration efforts (Bernhardt et al., 2005a; Palmer et al., 2005). Given such uncertainties, it seems unlikely multiple restoration projects will necessarily provide consistent, additive water quality benefits across space or through time. This is an operating assumption yet to be evaluated across an entire watershed, however. Even so, it remains unclear whether the benefits of riparian system restoration result from nutrient interception (Lowrance et al., 1997; Weller et al., 1998), improving stream uptake potential via restoration of stream functionality (Peterson et al., 2001; Bernhardt et al., 2005b), reducing pollutant loadings by removing land from production (Dosskey, 2001), or some combination of these alternatives focused on the needs within specific landscapes. Understanding the spatial effects of these management alternatives and their potential benefits should allow greater definition of coordinated monitoring strategies and more effective prioritization of restoration spending.

Cumulative effects of economics, policy, land ownership, and ecological recovery

Complex interactions between land uses, economics, policies, and ecological processes strongly influence the timing of physical, chemical, and biological responses to conservation practices. Land use patterns reflect aggregate outcomes of rational decisions by individual landowners to optimize returns from their agricultural resources, but discrete priorities by landowners rarely result in ecologically well-integrated watersheds. Political policies affect land use change more rapidly (2 to 20 years) than the ecological processes (10 to 100 or more years) we are trying to conserve or restore. As a result, most agricultural landscapes exhibit spatial patterns of land cover and aquatic and terrestrial communities that primarily reflect the "footprint" of impermanent policies and short-term economic decisions.

Landowners, communities, and resource managers are always faced with choices of actions that sustain, deplete, or rebuild existing resources (Pitcher, 2001). Industries and societies that harvest or extract natural resources often observe gradual, long-term depletion of environmental assets. Pitcher (2001) identified three major tendencies of fisheries harvest that tend to cause a "ratcheting effect" leading to resource depletion. The first depletion effect, which he termed "Odum's ratchet," is the tendency for past ecological conditions to become harder to restore when species (or genotypes) become extinct. As we lose biological components, ecological functions are more likely to be irreversibly changed.

The second depletion effect, termed ''Pauly's ratchet,'' is based on the tendency for each of us to relate changes in our ecosystems to what those systems were like when we began our careers. "Accounts of former great abundance are discounted as anecdotal, methodologically naive, or are simply forgotten" [Pauly (1995), as quoted in Pitcher 2001].

The third depletion effect, termed ''Ludwig's ratchet,'' is the tendency to increase harvest capacity through financial investment that requires continued amounts of declining resources to be harvested, generating further investment in technological capacity to harvest more resources.

Agricultural parallels are obvious, such as increased crop productivity leading to soil, water, and nutrient depletion, which requires loans for more specialized equipment and agrochemicals, which requires sustained production to repay loans required for their purchase, resulting in increased harvests from systems where soil and water resources are already becoming increasingly limited. Just as ocean fisheries have witnessed serial depletions within and among species caused by overharvest as a consequence of technological advancements in the fisheries industry, agriculture has experienced shifts in crop types and land uses as agronomic capacity becomes altered and required resources become scarce (Potter, 1998; Cochrane, 2003).

In light of the dual nature of conservation and restoration, an additional ratchet effect—"the restoration ratchet," can be added to those defined by Pitcher. This ratchet mechanism reflects the tendency to view conservation and restoration as immediately and fully effective, thereby offsetting choices leading to more intensive land use, further depleting remaining resources. In reality, the outcome of restoration may not be realized for decades after it is first implemented, and the success of conservation of existing resources remains largely unproven. This inherent tendency to assume efforts to restore depleted resources immediately counterbalance actions that deplete resources inevitably leads to continued decline in natural resources as well as ecosystem structure and function.

Achieving greater conservation effectiveness at landscape or watershed scales

Timeframes for responses to restoration actions

Realistic timeframes for responses to ecological restoration in agricultural landscapes can be rapid (1 to 5 years), relatively fast (5 to 20 years), intermediate (20 to 50 years), slow (50 to 100 years), or extremely slow (greater than 100 years). Why do ecological processes and ecosystem components exhibit such widely differing rates of responses to restoration efforts? Many factors contribute to the timing of responses of different landscape structures, populations, and communities. Agricultural landscapes contain complex physical landforms, chemical environments, biotic communities, human communities, and histories of change. The characteristics of all of those fundamental features of agricultural land vary enormously from location to location. Therefore, it is impossible to identify exact timeframes for ecological responses to restoration efforts. We summarize several factors that shaped the responses observed in the examples we presented in Table 2.

The landscape and its physical processes set limits on potential rates of recovery in terrestrial and aquatic systems. For example, many river channels throughout the United States have been simplified and straightened. Restoration of river channels requires reconnecting historical side channels and floodplains, reestablishing channels where they have been eliminated, and restoring natural flow regimes to the extent possible. The rate of recovery of those channels will depend upon the occurrence of natural flood processes that shape and maintain river channels and their floodplains. Timing of such restorative floods will depend upon the chances of their occurrence and future weather patterns.

Rates of ecological recovery also depend upon the degree to which the system has been altered. Obviously, a slightly altered system is likely to recover much more rapidly than a landscape that has been greatly changed. For example, a farmland with large patches of native forests and relatively well connected riparian forests will respond rapidly to restoration efforts that reconnect the fragmented pieces. In contrast, a farmland that is almost completely converted to cropland, with little or no remnant native forests, will require 50 to 100 years or more to begin to support native terrestrial and aquatic communities endemic to native forests.

Recovery of ecosystems depends upon the availability of species and the resources they require. As a result, the legacy of past systems can influence recovery. For example, old-growth forests develop diverse microbial communities and organic matter in their soils. In the decades following harvest of old-growth forests, the soils contain organic matter, microbes, seeds, and invertebrates from the old forest. After repeated harvest cycles, organic matter becomes depleted, microbial diversity declines, and invertebrate communities shift to those adapted to earlier stages of forest succession.

Legacies are also important in terms of contaminants and nutrients applied and accumulated over time in agricultural landscapes. Legacies of contaminants can cause recovery to be extremely slow. Contaminants that breakdown slowly and are strongly attached to soils and particles may reside in agricultural soils for decades after agricultural practices change. The long-term trend in the persistence of DDT is an example. DDT breaks down to other chlorinated forms of hydrocarbons in 5 to 10 years, but the other forms (DDD and DDE) commonly are found in soils, organisms, and water for 30 years or more. Some chemicals, such as heavy metals like mercury and arsenic, can bind to soils and remain in storage for centuries.

Rates of ecological processes create limits for recovery. One obvious example is riparian shade. When restoration programs plant native trees along streams to restore shade, it is obvious that seedlings will provide little shade. Several decades (20 to 50 years, depending upon species) may be required to develop full canopies. If the project goal includes restoration of amounts of large wood in streams, more than 50 to 150 years may be required before the streamside forests begin to deliver wood to streams.

The recovery of populations depends upon rates of birth and death. Species that reproduce rapidly and produce large numbers of offspring may recover quickly after restoration is implemented. In contrast, species that reproduce and mature slowly and produce low numbers of offspring will require much longer (decades to centuries) to recover.

Table 2. Factors that deterimine the timeframe for responses to restoration efforts.

	Recovery period			
System attributes	**1–10 years**	**10–50 years**	**50–100 years**	**100–1000 years**
System complexity	Simple	Simple	Complex	Complex
Control of inputs	Simple to control	Simple to control	Difficult to control	Difficult to control
Flow paths	Rapid	Intermediate	Slow	Very long and slow
Storage of nutrients, toxics, sediments, or human additions	Low	Moderate	High	High
Reproductive rates of native biota	Rapid	Rapid	Slow	Slow
Required stages of succession	Succession not required	Early stages	Mature stages	Late stages
Legacies of native ecosystems	Abundant	Abundant	Few	Few to none
Influence of alien species	Little	Slight	Extensive	Extensive and dominan
Degree of landscape alteration	Minor	Intermediate	Major	Major and irreversible

If land use practices causing ecological degradation continue after restoration efforts, recovery will not occur as rapidly. The degree to which pressures are placed on the recovering resources determines the rates of recovery. For example, some restoration of riparian areas involves establishment of livestock grazing exclosures. Such exclosures may encompass complete exclusion of livestock grazing or limited seasonal use. The amounts and timing of grazing can greatly influence the rates and degree of riparian and aquatic system recovery.

Couplings between the physical landscape and biological communities take time. Floodplain restoration requires reestablishment of periodic inundation. In turn, this results in changes in sediment deposition and channel change. In response, floodplain vegetation can be altered, and the composition of plant communities shifts through time as succession occurs. In turn, future floods interact with developing floodplain forests, changing the patterns that developed previously. Such interactions can proceed for decades, and outcomes of restoration efforts will reflect these changes.

Future directions to make restoration more effective

We have explored several fundamental temporal perspectives of ecological responses to restoration and conservation practices. But the larger question is how can communities and natural resource agencies become more effective in the conservation and restoration practices applied to agricultural landscapes? We suggest six major approaches that offer substantial promise to create more effective conservation and restoration: (1) Greater consideration of producer/landowner attitudes and knowledge, (2) more effective in-field practices and planning, (3) greater emphasis on effective monitoring and assessment, leading to refinement of policies and practices, (4) adoption of landscape perspectives in planning and applying conservation practices, (5) development of conservation markets, and (6) expansion of the use of alternative future scenarios.

Producer/landowner attitudes and knowledge

Agriculturalists value the culture, environmental worth, and aesthetic characteristics of their land, but personal opinions on the values of natural amenities vary. Often, one person's wildflower is another's weed. For the most part, however, those involved in agriculture embrace a desire to improve the quality and productivity of land to be passed on to future generations (Lubchenco, 1998; Wildlife Management Institute, 2006). Management philosophies guiding contemporary agricultural land use have evolved largely on the perception that composition, diversity, and ecological relations between farmed and nonfarmed land play only a small, if any, roll in productive agricultural systems (O'Riordan, 2002; Kirschenmann, 2003; Keeney and Kemp, 2004). Agricultural ecosystems are no less complex than any other ecosystem. Variability in frequency and types of land use, diverse goals of landowners, skepticism about outside intervention in management decisions, and suspicions about regulation contribute additional layers of complexity in addressing environmental issues associated with agricultural land use.

The effectiveness of conservation programs is ultimately defined by the willingness of landowners to participate and their knowledge of conservation practices and their benefits. Long-term solutions to entwined issues, such as soil erosion, water quality, and wildlife habitat, will be achieved only when conservation policies are embraced across multiple farmsteads to the watershed level. Incorporation of landowner knowledge about local issues and production challenges, coupled with forethought directed to their expectations and limitations, will elevate interest and create opportunities to improve the level of landowner knowledge required for successfully implementing conservation practices and programs. The most proficient way to get information to farmers about the benefits of conservation is to have it delivered by a neighbor who has seen success. This can then be followed up with educational activities to improve landowners' abilities to implement conservation practices successfully.

Large-scale assessments of conservation effectiveness based on sophisticated modeling are necessary for understanding effects of and refining conservation policies. Such approaches rarely, however, furnish site-specific answers to those who have invested time, labor, and trust in adoption of conservation practices on their farm. Approaches for describing on-farm or within-watershed effects of conservation are needed to strengthen and justify program participation. Many landowners who enroll in conservation programs value the environmental benefits associated with their conservation activities and want to know how well conservation practices are working on their farms. Some landowners are willing to participate in the collection of information needed to describe effectiveness of conservation policies (Wildlife Management Institute, 2006). Programs such as the Izaak Walton League's Save Our Streams (Izaak Walton League of America, 2006), where landowners are trained in sampling and identification of aquatic insects to estimate changes in water quality brought about by adoption of conservation practices, can serve as models for involving willing landowners in monitoring conservation effectiveness. Identification of specific, regionally important species as management and monitoring priorities, addressing effects of conservation practices on multifarm or watershed scales, consideration of landowner goals/limitations, as well as finding ways for willing landowners to become part of monitoring activities will improve abilities to furnish meaningful results needed to refine the performance of agricultural conservation programs.

Innovation in farm operations and waste management systems

A key factor in conservation practice effectiveness is timely adoption. Practices that are simple, easy to implement, and fit well into the agricultural operation are those most likely to be adopted by a significant number of producers. Use of precision agriculture, nutrient management, integrated pest management, on-site wastewater treatment, improved buffer designs (e.g., carbon-source trenches for enhanced denitrification), and improved livestock nutrition to reduce nitrogen and phosphorus in manures offer potential to increase the effectiveness of future restoration efforts. More specific innovations include the following:

- Use of existing in-field conservation practices (nutrient management, integrated pest management, conservation tillage, etc.) that reduce production costs and reduce resource loss from the field.
- Targeting implementation of conservation

practices by identifying critical source areas within fields or landscapes.
- Elimination of agricultural subsidies that distort costs and encourage producers to over apply agricultural chemicals and farm marginal land that would not otherwise be economically productive.
- Implement conservation programs and practices that measurably improve the environment rather than those only presumed to protect the environment.
- Evaluate and improve success of conservation programs/activities by measuring improvements in environmental quality.
- Fund only conservation programs and activities that have explicit, measurable environmental goals.

Assessment and monitoring

Given the large investments of public funds in conservation and restoration actions, any prudent society would want to determine whether its efforts are successful. But observations and assessments of conservation program performance require a commitment of effort and funds. Because so few restoration programs are monitored, little information feeds back into the policy formulation and decision-making processes. As a result, adaptive management occurs most often through sequential but disconnected correction measures or emergence of new programs. A recent review of river restoration projects found 20 percent had no defined objectives, and only 10 percent included any form of assessment or monitoring (Bernhardt et al., 2005a). Post-project assessment often focuses more on implementation (e.g., how many acres or stream miles have been treated) rather than achievements of intended environmental goals, such as measurable reduction in agricultural chemicals entering surface waters.

One of the major reasons for the low rates of monitoring and assessment is the relative cost of restoration actions versus monitoring and assessment. Most people and agencies are well intended and want to invest as much as possible in actual restoration activities. As a result, few projects dedicate funds and effort to determine whether the projects are truly successful in meeting environmental objectives, trusting that implementation of the practices alone meets program goals.

For many projects, the timing of monitoring is poorly matched to realization of expected responses. A familiar example is planting of riparian vegetation intended to reduce soil erosion, increase bank stability, increase shade, lower stream temperature, and enhance water quality, as well as the abundance, diversity, and health of fish, wildlife, and other organisms. Typically, such projects are evaluated for two to five years after establishment to determine survival of the planted vegetation. In that two- to five-year interval, it is unlikely the plant communities could develop to a stage in which they provide the intended ecological contributions (e.g., canopy cover, food inputs, wood, channel complexity). Twenty to 50 years or more is a much more realistic time horizon for recovery of many of these ecological functions.

Nonetheless, many involved in restoration understand the long-term nature of the process. In a survey of Pacific Northwest watershed councils, Bash and Ryan (2002) noted, "Many respondents indicated that short-term project assessments might not be meaningful given the time frame needed to evaluate the outcome of restoration projects."

It is highly unlikely that funds and workforce will ever be adequate to monitor and assess a large portion of the conservation or restoration actions on agricultural land. One option to provide rigorous assessment of conservation and restoration is creation of a "monitoring bank." Various projects throughout a region, from a variety of sources, could invest in a common fund that would support scientifically rigorous assessments of the major conservation and restoration actions applied to agricultural land in the region. Sites could be randomly selected from a systematic database, with factors being measured or monitored that reflect the greatest need for information identified by a ranking process that includes priorities from all agencies contributing to the monitoring bank. Conclusions drawn from study results could be scaled appropriately to the spatial and temporal scales that reflect regional applications for the intended conservation and restoration outcomes. Such an approach would eliminate duplication of monitoring efforts and maximize results from funds allocated for monitoring and assessment of conservation and restoration practices for all agencies involved in the program.

Achieving conservation effectiveness at landscape scales

The multiscale nature of watershed processes requires a watershed approach to management, but effective management of watersheds is challenging in landscapes under multiple ownerships (Allen el al., 1997). NRCS provides technical assistance to develop comprehensive resource management systems on land that may or may not be involved with conservation-oriented management. Practices implemented within the framework of a resource management system effectively protect soil and water quantity and quality as well as associated terrestrial wildlife communities. With such practices in place, aquatic species are also likely to benefit. Sedimentation of streams causes damage to habitats of all aquatic species, but that damage can be diminished when beneficial land management practices are implemented at broad scales (Lenat, 1984) and coupled with riparian conservation practices at smaller scales (Stauffer et al., 2000). Indices of biotic integrity provide insight on the effects of these practices on aquatic fauna at both scales (Lammert and Allan, 1999; Weigel et al., 2000).

Finding collaborative ways for landowners to maintain or restore connectivity of habitats should contribute to ecological restoration across wider geographic scales. For example, use of "best development practices" to improve the trajectory of amphibian populations has showed promising results when implemented cooperatively at the town level in Vermont (Calhoun et al., 2005). Maintaining contiguous riparian zones, or buffers, of adequate width along streams and rivers has been shown to correlate highly with improvements in indices of biotic integrity for aquatic fauna in Wisconsin (Weigel, 2003).

Conservation markets for regional communities

Assessments have traditionally overlooked the economic gains that can result from adoption of conservation practices. Mitigation banking has been used most widely to provide conservation benefits while also creating economic opportunities. Pollution trading is emerging as a major economic choice in response to TMDLs and other regulatory criteria. Also, state agencies are beginning to implement conservation payments to offset consumer impacts (such as large sport-utility vehicles). All of these create opportunities for farmers to implement conservation practices that potentially increase their income.

Focus on future demands and challenges rather than past practices

All too often regional assessments of conservation and restoration focus on examination of ecological conditions assumed to be related to past and current land use. Rarely are potential consequences projected for the near future (approximately 50 years). Consequently, as problems of the past are addressed, management typically fails to anticipate future challenges. Emerging resource issues are repeatedly addressed with tools designed to repair the consequences of past land use and management practices. Immediate or short-term responses often are considered to have greater likelihood of success; they often are perceived as being more credible and defensible than accepting the risk of addressing unknown changes in policy and management that might potentially affect long-term changes in resource availability or environmental conditions. As a result, decisions tend to favor near-term choices affecting small, local areas.

A proactive, longer term tool potentially applicable to management of agricultural landscapes is assessment of alternative future scenarios. Alternative-futures analysis has been used to explore future trends in the Willamette River Basin (Baker et al., 2004), as well as the San Pedro River in Arizona and Camp Pendleton in California (Steinitz et al., 2003, 2005). These assessments of future trends provide spatial projections of alternative choices about land uses and the potential environmental, economic, and social consequences of those alternatives.

A study of future alternatives for Arizona's San Pedro River demonstrated that availability of water will have the greatest impact on future ecological conditions in this arid region (Steinitz et al., 2005). Irrigation withdrawals were projected to have the greatest potential impact on ecological processes, but policies that encouraged population growth and relaxed constraints on development also would have major impacts on water and ecological conditions. A study of land use alternatives in the upper Midwest examined people's choices for residential development in an agricultural region, finding that a majority preferred landscapes with natural vegetation and higher ecological conditions (Nassauer et al., 2004). Though questions of rural land conversion remain, the communities clearly view an ecologically healthy landscape as a more livable environment.

Environmental changes under alternative futures can be evaluated quantitatively through simulation models or observed relationships and qualitatively through expert judgment or the Delphi approach (Hulse and Gregory, 2001; Hulse et al., 2002). Mechanisms for identifying assumptions and spatial representation of alternative future scenarios are just as important, however, as are methods for analyzing alternative futures. Three approaches have been used in recent years—stakeholder-derived, expert-derived, and model-based scenarios. Each approach has strengths and weaknesses. Stakeholder processes employ citizen stakeholder groups to define assumptions about how future land and water use will unfold. Those scenarios can be used with planning processes and models to produce maps of potential future land and water use, translating the stakeholder assumptions into mapped form. The stakeholder approach has the advantages of citizen involvement, greater political plausibility, and an increased likelihood of institutional acceptance. But stakeholder–driven processes have one disadvantage: They do not statistically quantify the likelihood of various alternatives, and the number of alternatives produced (three to ten) is typically limited.

A second common approach for creating mapped alternative futures is expert judgment, with professionals in the biophysical and social sciences defining processes and rates of transition that may determine future land and water use conditions. Alternative futures produced from expert judgment have the advantage of quantifiable statistical likelihood (based on the larger number of alternatives produced), but suffer from unclear political plausibility and a lack of citizen involvement, which often limits their credibility in affected communities.

Simulation modeling has been used to define alternative futures by representing the rules by which people make decisions and then projecting probable effects across the landscape. Simulation models can produce thousands of possible future landscapes, with the advantage of representing the statistical likelihood of various alternatives. An additional advantage of simulation models is the ability to create and run new alternatives quickly.

Trajectories of land use and environmental change from 1850 to 2050 were developed for the 30,000-square-kilometer (11,583-square-mile) Willamette River Basin in Oregon, a basin comprised of approximately 25 percent agricultural land, 65 percent forest land, 6 percent urban land, and 4 percent rural residential land (Baker et al., 2004). Human population in the basin is expected to increase from 2.2 million to more than 4 million by 2050. Three spatially explicit future scenarios were developed by a group of stakeholders: (1) a Plan Trend 2050 scenario in which current policies and practices continue through 2050, (2) a Development 2050 scenario in which market forces are allowed to influence land use change and current land use policies are relaxed, and (3) a Conservation 2050 scenario in which additional, plausible conservation and restoration practices are implemented. Scenario outcomes were evaluated on the basis of land cover change, water availability, and models of ecological conditions for fish, macroinvertebrate, and wildlife communities.

Incorporation of conservation practices in Conservation 2050 enhanced wildlife habitat without significantly altering the function of the agricultural system. Development 2050 also showed local improvement in wildlife habitat due to increases in natural vegetation associated with the developed environment. Plan Trend 2050 indicated little change in habitat quality because few modifications were made to agricultural land.

The Willamette Valley contained approximately 240,00 hectares (620,000 acres) of prime farmland in 1990, almost all of which remained in agricultural production (1 percent was converted). Under the Development 2050 scenario, approximately 25 percent of prime farmland would be converted to other uses, leading to fragmentation and conversion of agricultural fields. Under the Conservation 2050 scenario, 15 percent of prime farmland would be converted to field borders, low-input crops in sensitive areas, and conversion of cropland to native vegetation. Development scenarios tended to prefer areas of prime farmland, while restoration activities tended to focus on lower quality, less productive farmland.

One of the most important findings of the alternative-futures analysis is that both the Plan Trend 2050 and Development 2050 scenarios show either little change or continued decline in natural resources (Figure 1). In sharp contrast, indicators of natural resource condition improve substantially under the stakeholders' assumptions about plausible restoration measures in the conservation 2050 scenario, recovering 20 to 70 percent of the losses sustained since settlement in the mid-1800s. Citizens and decision-makers in the basin now have geographic projections over the next 50 years, indicating conservation and restoration practices are likely to produce significant ecosystem benefits while accommodating the projected increase in the human population.

An agent-based model, Evoland (Evolving Landscapes), was developed to examine ecological and economic consequences of alternative futures for floodplains and riparian areas of the Willamette River Basin (John Bolte, Oregon State University, personal communication). This modeling approach allows rapid analysis of many alternative futures, measurement of variance based on probabilities of land use choices, and modification of assumptions and policies defined by user groups. Results of modeling alternative policies clearly demonstrate conservation and restoration policies can be effective in restoring ecological function in the long run (20 to 40 years), but ecological conditions respond to conservation and restoration actions more slowly than they do in response to economic and social policies. An additional concern raised focuses on effectiveness of adaptive management. If policies were implemented that would result in short-term economic gain, but cause floodplain and riparian degradation not evident for 20 to 30 years, adaptive management would be ineffective in the face of the substantial financial investments that would have occurred before the undesired outcomes were realized. The timing of restoration outcomes will be constrained by the competing processes of intensified land use and land use conversion.

Making decisions for generations

In his 1999 book *The Clock of the Long Now*, Stewart Brand addresses the challenge of incorporating different time scales into the decision-making process. He asks, "How do we make long-term thinking automatic and common instead of difficult and rare, and how do we make the taking of long-term responsibility inevitable?" Tools and ways of thinking have to be changed so that the "long now" is inherent in the management questions asked and the solutions explored. Because environmental and social consequences of modern agricultural production reach from the heart of this continent into coastal and marine ecosystems, we can no longer measure agricultural accomplishments simply on economic returns brought about by traditional farm products.

Unfortunately, there has been an inclination to define the agricultural landscape as being composed of either "working" or "conservation" land. This regrettable distinction, born in part by the structure of existing conservation programs, fails to recognize that all land, regardless of production status, is part of the "working" agricultural ecosystem. An economically viable, environmentally sound, and, therefore, socially supportable agricultural industry will be possible only when agriculture protects and even perhaps enhances the natural and cultural resources upon which it stands.

Budgetary constraints increasingly force decisions affecting how conservation programs are designed and administered. Successful conservation policies can be publicly and politically supported only when their effectiveness is known. To gain such knowledge requires an unrelenting commitment to calculate both immediate as well as long-term effectiveness of programs and refine conservation policies as information becomes available. The reality that must be faced is hard numbers that either support or disprove the success of conservation and restoration activities will not appear quickly, nor will they come without a commitment to fund the research required to define acceptable solutions.

Traditional conservation and restoration practices will continue to be used. Well-intended landowners and community groups will continue to try to sustain and restore declining resources in the face of growing human populations and their need for agricultural commodities. There will be no easy answers, and good intentions alone will not suffice. The tremendous power of the ratchet effects in place in society—extinction of species, generational views of resource abundance and landscape condition, and economic pressures that require continued or accelerated commodity production—must be faced. And through the

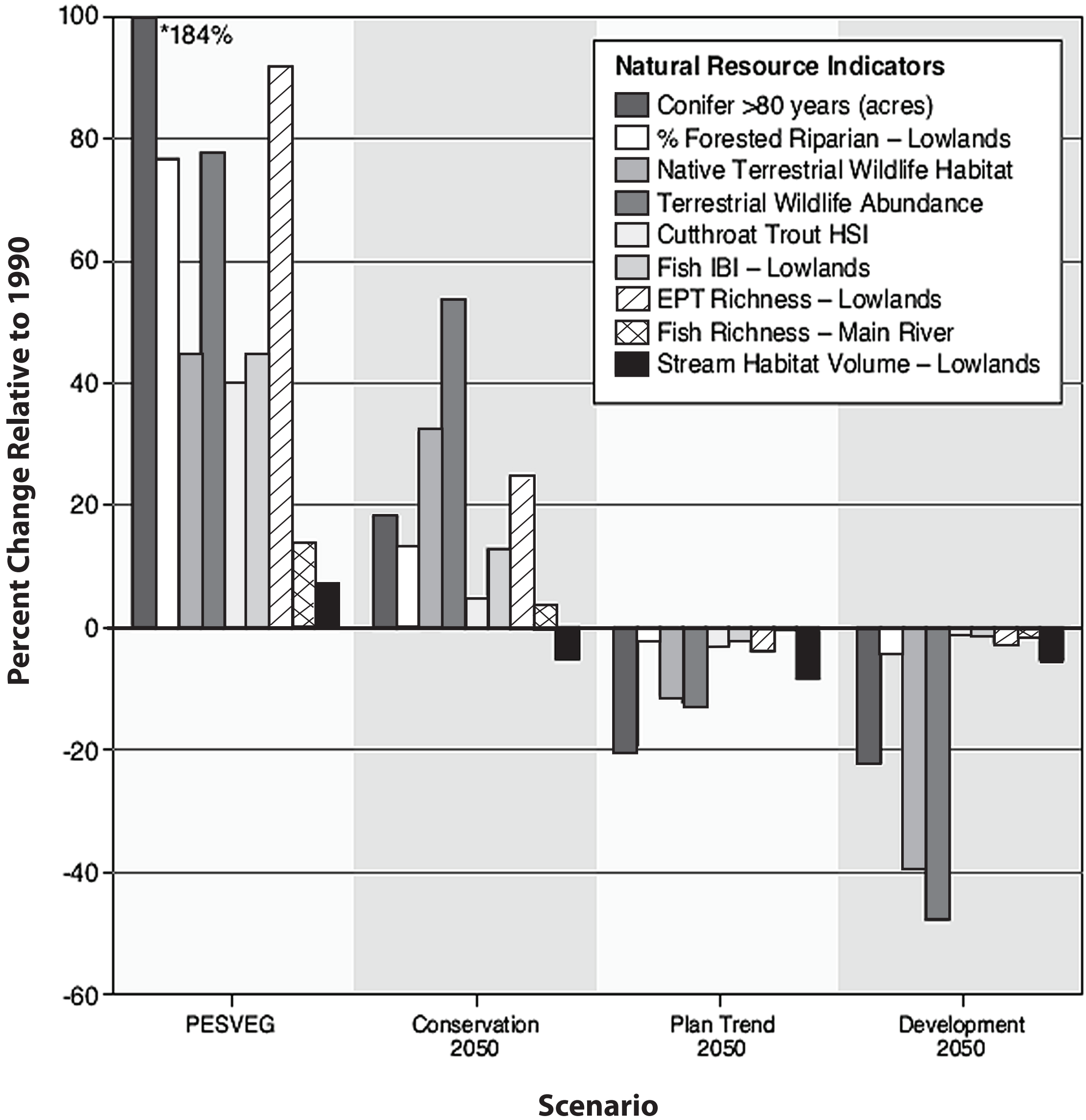

Figure 1. Percent change in measures of natural resource condition in the three future and pre EuroAmerican scenarios relative to 1990 land use and cover in the Willamette River Basin. Source: Baker et al., 2004.

temporal perspectives explored in this paper, the timeframes of both conservation and restoration must be carefully and clearly explained to avoid the ratchet effect of assuming these practices will be immediately and fully effective. It cannot be assumed that continued or accelerated demands on natural resources can be counterbalanced by conservation and restoration measures alone. The uncertainty in that assumption, even when balanced with more realistic expectations of timeframes, must be adjusted with a “margin of safety” for natural resources, just as engineers would use in designing any road or building. Landowners and resource managers must balance the immediate impacts of their actions against the current rates of resource restoration. The actions taken today determine the extent to which the world will sustain the next generation. People today owe it to the next generation to base today’s decisions on realistic expectations about practical timeframes for achieving ecological restoration and conservation.

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Roundtable: Realistic expectations of timing between conservation and restoration actions and ecological responses

Roundtable participants engaged in a wide-ranging discussion on many topics, most of them at least somewhat related to "realistic expectations." Among those topics were the following:

- Responses to environmental degradation are often technological fixes, but are the right end points being measured? The baseline of the "healthy" condition is often not known. Reversion to pristine conditions is impossible. "Recovery" is a healthy, diverse ecosystem, not native condition. The nitrogen cycle is distorted; the phosphorus cycle is broken; and hydrology has been altered. Balance and quality control are needed, but the economics does not work out. The Chesapeake Bay project was offered as an example.
- Ecological trajectories must be assessed to determine where they will lead in the future. The historical context is an important starting point from which to look forward and understand the trajectory of change. Factors must be assessed that cause changes in trajectories; what-if scenarios must be examined; and future scenarios from models must be developed. Population growth and pressure must be considered in these scenarios, along with climate change.
- How can people relate to realistic expectations? Realistically project or even come up with expectations? What limits what we can realistically expect or how we can change expectations? What directs the evolution of value systems through generations? Political will is needed to bring "realistic expectations" to reality, perhaps more than scientific or stakeholder interests.
- In considering expectations, the focus must be on progress—the right trajectory—rather than just end results.
- How should understanding of effects and expectations be scaled up from individual fields to entire watersheds?
- Agencies and other institutions continue to be data rich and information poor. Scientists could help by sorting out the key questions that might help turn existing data into useful information.
- Public involvement and sorting out what the public wants for the future is important. Community visioning processes and other exercises that help identify what is realistic and believable can help. The costs and behavioral changes involved need to be included in these processes.
- Policymakers must realize that conservation and restoration are long-term processes. Meaningful responses to conservation cannot be expected in the time frame of individual farm bills (five to seven years). Conservation effectiveness will require much longer time frames.
- Regional priorities must be defined that are meaningful to local farmers and populations. Environmental goals that are unrealistic and do not support reasonable integration of conservation and viable continuation of agricultural land use will not be accepted by farm operators.
- Monitoring of conservation effectiveness must be part of all conservation programs. A relatively small amount of high quality data can be used to extrapolate results to much larger

areas and programs. But program managers must have the data, budgets, and long-term commitment to collect such information.

- Long-term monitoring of the effectiveness of agricultural conservation will require not only provision of financial support but development of an infrastructure that will support long-term collection of useable data and results. This will require setting measurable, reasonable goals and identification of an agency responsible for training, data quality control, interpretation of results, and getting those results to the public and people who make long-term agricultural policy decisions (U.S. Department of Agriculture officials and political representatives in Washington, D.C).

Roundtable participants then reached consensus on a series of leading questions that at least implied what the most important next steps might be in strengthening the science important to agricultural conservation:

1. How do we identify reasonable expectations? How do we communicate them to the public and policymakers? How do we receive communications back from the public and policymakers? How do we make adaptive management work in the "real world," that is, how do we involve the public in adaptive management (and who are "we")?
2. How can we develop a process to identify and influence trajectories of change and do so at an ecosystem/landscape level rather than a localized, single-issue level? What are the costs and benefits of alternative trajectories? There are many measures to assess in evaluating alternative future scenarios. A process for doing this has been used in some areas, but is not widely available or widely known.
3. What questions do we need to ask and answer to turn data into information that can be used to refine realistic expectations? Where do we need more data, and where do we just need to analyze what we have?
4. What is an appropriate timeframe in which to develop reasonable expectations? What are people's/politicians' typical timeframes? How do "realistic" expectations change when the time frame is 2 to 4 years, 10 years, a generation, 100 years, or more?
5. Realistic expectations are subject to change over time. What factors, both catastrophic and evolutionary, cause perceptions of what is realistic to change? What can we do to avoid being only passive participants in this process?